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Biochar use in forestry and tree-based agro-ecosystems for increasing climate change mitigation and adaptation

Ilan Staví*

Dead Sea and Arava Science Center, Ketura, 88840, Israel

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This study reviews the potential use of biochar as soil amendment in afforestation, reforestation, agroforestry, fruit tree orchards, and bio-energy plantations. Implementing this practice could sequester large amounts of carbon (C) over the long-term, potentially offsetting anthropogenic emissions of carbon dioxide, and mitigating climate change. On a global scale, this practice could sequester between 2 and 109.2 Pg biochar-C in 1.75 billion ha of degraded and deforested lands and agroforestry systems. An additional considerable amount could be sequestered in the soil of fruit tree orchards and bio-energy plantations. The anticipated improvement in the quality of the biochar-amended soils is expected to enhance resilience to these land uses, increasing their adaptation capacity to climate change. Yet, specific questions still need to be addressed, for example, the impact of biochar on the availability of nitrogen and potassium for plants in acidic soils and under humid conditions, as well as the impact of biochar on soil and trees in alkaline soils and under Mediterranean or drier conditions. Also, a full assessment of health hazards and environmental risks related to the production of biochar and its application in soil is required. Other questions relate to the environmental and economic costs of biochar application. Therefore, life cycle assessments and economic calculations should be conducted on a site-specific basis and include the practices of feedstock collection, transportation, processing, and spreading. International actions should regulate biochar practice as an eligible means for funding under the C finance mechanism. Specifically, payments should be provided to landowners for accomplishing ecosystem services.

Keywords: carbon sequestration; Clean Development Mechanism (CDM); environmental regulations; land restoration; Reducing Emissions from Deforestation and Degradation (REDD)

Introduction

Forests cover almost 4 billion ha or 30% of the global land area, and are considered the major terrestrial regulator of the carbon (C) cycle. Natural forests are widespread throughout several biomes, including the equatorial, tropical, Mediterranean, warm temperate, temperate, and boreal (Easterling et al. 2007; Nabuurs et al. 2007). The natural forests are generally described as evergreen broadleaf, evergreen needleleaf, deciduous broadleaf, deciduous needleleaf, or mixed (De Fries et al. 1998). A report published by the FAO estimated global forests’ C pools as 221 Pg in above-ground biomass and 61 Pg in below-ground biomass, yielding together a total biomass-C pool of ∼282 Pg. According to this report, global forests’ C pool in dead wood, litter, and the upper 30-cm soil depth accounts for 39, 25, and 287 Pg, respectively. Therefore, the total forests’ C pool throughout the globe was estimated in 2005 as ∼633 Pg (Marklund & Schoene 2006). In undisturbed forests, the C balance has a tendency toward slight accumulation as its uptake through photosynthesis exceeds losses from respiration. However, net C uptake is offset by disturbances, such as wildfires, droughts, or diseases, which periodically lead to substantial losses from both vegetation and soil. Also, anthropogenic activities often function much like natural disturbances in their effect on a forest’s C balance (Conant 2011).

Land-use change results in a considerable decrease in forest area. Between 2000 and 2005, the global deforestation rate was 12.9 million ha year−1, mainly as a result of converting forests to agricultural land, but also due to the expansion of settlements, infrastructure, and unsustainable logging practices (Nabuurs et al. 2007). Forest degradation could be somewhat reversed through implementing forestry projects. In 2005, intensively managed forest plantations comprised ∼4% of the global forest area. The area of this land use has been rapidly growing, with an annual increase of between 2.5 (Easterling et al. 2007) and 3.5 million ha (Niles et al. 2002). Due to forestry efforts, landscape restoration, and natural expansion of forests that, to some extent, reverse the total loss of forest lands, the most recent estimation of global net loss of forest is 7.3 million ha year−1 (Nabuurs et al. 2007). A recently published synthesis study compared the C sequestration capacity of primary and secondary natural forests to that of afforestation and reforestation plantations. Overall, compared with the natural forests, the plantations were reported to have 11% less above-ground biomass. Also, fine root biomass, soil organic C (SOC) concentration, and soil

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microbial C concentration were reported to be, respectively, 66%, 32%, and 29% lower in plantations than in natural forests (Liao et al. 2010). In addition, Liao and colleagues reported 22%, 20%, and 26% smaller available N, P and K, respectively, in the soil of plantations than in the natural forests.

The productivity of forests and climatic changes are closely interrelated. It is foreseen that the magnitude, frequency, and duration of extreme climatic events will increase as the Earth’s atmosphere warms. From 2001 to 2010, global temperatures averaged 0.46°C above the 1961–1990 average. Recent warming has been especially strong in Africa, parts of Asia, and parts of the Arctic. Temperatures recorded during 2001–2010 in the Saharan-Arabian region, east Africa, central Asia, Greenland, and Arctic Canada have all been 1.2–1.4°C above the long-term average and 0.7–0.9°C warmer than in any previous decade (WMO 2011). It is acknowledged that the rising concentrations of atmospheric carbon dioxide (CO₂) are expected to have a ‘fertilization effect’ on forests, potentially increasing their production capacity (Beedlow et al. 2004). However, forest C sinks are unlikely to persist indefinitely, and therefore, continued warming will probably diminish and potentially even override the fertilization effect (Gullison et al. 2007). Together with increasing air pollution and decreasing nutrient availability for plant uptake, the total impact on forests productivity is unclear. Global warming is also expected to exacerbate risks to forests from ozone (O₃), because hot weather and high atmospheric pressure promote its formation. Moreover, increased fossil fuel use will probably increase the production of O₃-forming air pollutants, resulting in decreased C sequestration capacity in biomass and soil (Beedlow et al. 2004). While a system’s productivity affects its ability to accumulate C, its resilience is largely the result of its capacity to cope with degraded physical conditions (Kandji et al. 2006). The outcome is a close association between the mitigation and adaptation capacities of forest ecosystems.

Biochar is the by-product of the C-negative pyrolysis technology for the production of bio-energy from organic materials. The process is conducted under complete or partial exclusion of oxygen (O) and relies on capturing the off-gases from thermal decomposition of the organic materials (Lehmann 2007). After application in soil, the high porosity of biochar and its negative charge increases the soil’s water-holding capacity and nutrient retention, augmenting availability of water and nutrients for plant uptake. Indeed, application of biochar in soil was reported to increase its physical and chemical qualities, resulting in greater net primary productivity (NPP) of the agro-ecosystem (Laird et al. 2010). At the same time, the recalcitrant nature of biochar considerably decreases its decomposition by microorganisms and enables the long-term sequestration of C in soil (Lehmann 2007). The biochar’s physical and chemical characteristics are determined by feedstock type and pyrolysis temperature. For example, higher salt and ash contents are expected in wheat straw than in wood-derived biochar (Kloss et al. 2012), and C content and nitrogen (N) content are greater in pine chips than in poultry litter-derived biochar (Gaskin et al. 2008). A higher pyrolysis temperature results in lower biochar mass recovery, greater surface area, elevated pH, higher ash content, minimal total surface charge (Novak et al. 2009), and lower cation exchange capacity (CEC) (Kloss et al. 2012).

Removal of volatile compounds at higher pyrolysis temperatures also cause biochars to have higher C content and lower hydrogen (H) and O content (Novak et al. 2009). The production of biochar is inexpensive and can be practiced by low-income populations (Stavi & Lal 2013). According to the International Biochar Initiative, 154 biochar projects were carried out in 43 developing countries in 2011. These projects took place within a range of operational and functional scales, including on household (26%), farm (41%), village (12%), cooperative (11%), and regional (10%) levels. Feedstocks contained a range of organic materials including timber mill waste, livestock manure, plantation prunings, field stover, and fruit waste. The biochar production scale varied from small to large, and the technologies varied from batch retort kiln to continuous process kiln (Kelpie 2011).

In spite of the several agronomic and environmental advantages offered by biochar, its production and use may impose specific environmental risks, as well as occupational and health hazards. For example, biochars can potentially contain toxic compounds, such as heavy metals, dioxins, and polycyclic aromatic hydrocarbons (PAHs). The occurrence of these contaminants in biochars is likely to derive from either contaminated feedstocks or the use of processing conditions that foster their formation (Verheijen et al. 2010). To some extent, the composition of PAHs can be controlled through modifying the pyrolysis’ temperature (Kloss et al. 2012). Moreover, it seems that tight control on the feedstock materials and pyrolysis conditions might substantially reduce PAH levels, as well as the emissions of dioxins and particulate matter associated with biochar production (Verheijen et al. 2010).

Despite the prevalent use of biochar in agriculture, very few studies have examined its utilization in forestry and other tree-based agro-ecosystems. Therefore, this paper aims to examine the possibility of augmenting C sequestration in forestry and other tree-based agro-ecosystems by applying this stable C-based soil amendment to soils. Also, since this management practice is expected to improve soil quality and fertility, this study examines its potential to increase forests’ resilience, or adaptation capacity, to climate change. In this regard, four major pathways were previously mentioned in which C emissions are mitigated through forestry activities: (i) an increase of forested land area (ii); an increase in the C density of existing forests (iii); expanded use of forest products to achieve a sustainable reduction of fossil fuel use; and (iv) reduction of emissions from deforestation and land degradation (Canadell & Raupach 2008). Either directly or indirectly, this review addresses each of these pathways. More specifically, the objectives of this review were (1) to raise awareness of the potential of biochar applications as...
a soil amendment in these land uses, in order to encourage research and development of this management practice; (2) to highlight the major challenges in implementing such projects; and (3) to demonstrate the need for international regulations to accomplish widespread implementation of this practice.

Deforestation and the C cycle

Recent estimations revealed that deforestation is the second largest anthropogenic source of CO₂ in the atmosphere after fossil fuel combustion. Cumulative emissions from deforestation and forest degradation account for between 12% (van der Werf et al. 2009) and 20% (Gullison et al. 2007) of total emissions. According to the more conservative estimation, these emissions yielded ∼1.2 Pg C year⁻¹ over the period 1997–2006 (van der Werf et al. 2009). Specifically, wildfires considerably alter forest characteristics through their impact on both trees and soil. Despite emitting huge quantities of CO₂ to the atmosphere, these fires also produce large amounts of ash and charcoal which modify the characteristics of the forest lands. Actually, wildfires provide important information about the impact of charcoal on forest ecosystems (Table 1). Due to the lack of studies on the topic of biochar in forestry systems, information of this type is crucial to increase the understanding of related processes. For example, in the northern boreal zone of Sweden, a greenhouse study that encompassed wildfire-produced charcoal mixed with three different substrates obtained from diverse microhabitats, revealed increased shoot-to-root ratio by Silver birch (Betula pendula Roth) and Scots pine (Pinus sylvestris L.). This effect was attributed to the greater uptake of N and other nutrients by trees that were presumably affected by the smaller binding of nutrients in the tree-litter phenolics in the presence of charcoal (Wardle et al. 1998). This is in accord with DeLuca et al. (2006), who reported that wildfire-produced charcoal (1%) mixed with ammonium (NH₄₉ 99%), increased nitrification rates and decreased solution concentrations of phenolics in Ponderosa pine (Pinus ponderosa Doug. ex. laws) litter. This is also in agreement with MacKenzie and DeLuca (2006) whose research in the same geographical region demonstrated that wildfire-produced charcoal was highly effective in sorbing litter-phenols of the understory elk sedge (Carex geyeri Boott) and ericaceous kinnikinnick (Arctostaphylos uva-ursi (L.) Spreng.). They also reported that charcoal increased net nitrification in the ericaceous litter, but not in the sedge litter. The long-term residence time of ash, charcoal, or black C has been widely acknowledged (Lehmann et al. 2008; Abiven and Andreoli 2011; de Lafontaine and Asselin 2011). Nocentini et al. (2010) discovered that the decomposability of charcoal is dominated by the size of its particles. In a study conducted in an Italian pine coastal forest, they found that in charcoal produced by a moderate-intensity wildfire, the charcoal fraction of <0.5 mm, accounted for 24% of the total charcoal mass, was rich in N and potentially susceptible to microbial decomposition. Nocentini and colleagues reported that wood-derived charcoal was largely prevalent in the largest fraction of >2 mm, while pine needles and herbs were more dominant in the smallest fractions. Santin et al. (2012) reported that extreme fires in mixed-species eucalypt forests and temperate rainforests in Victoria, Australia, converted tree-biomass into ash, resulting in deposits of between 4.8 and 8.1 Mg C ha⁻¹ on the forest floor. Yet, while some of the ash remained on-site, much of it was subsequently redistributed, accumulating in foot slopes and depressions, or entering fluvial systems and forming a part of aqueous deposits. Also, Wardle et al. (2008) reported that during a 10-year study of the Swedish boreal, the loss of forest humus increased when mixed with charcoal through either respiration or leaching of soluble compounds. They proposed that, despite long-term sequestration of charcoal-C, it could be partially offset by stimulating the loss of plant litter-C. However, after a much shorter period – 240 days – Abiven and Andreoli (2011) reported no impact of charcoal on decomposition rates of different organic substances obtained from a mixed forest in Switzerland, which had been combined with the upper mineral horizon of a cambisol. Therefore, it seems that the impact of charcoal on C dynamics of surrounding organic substances depends on their nature, the soil conditions, and temporal duration.

Deforestation increases the impact of climate change on the remaining forests. Global warming puts tropical forests at risk of more frequent and severe droughts (Gullison et al. 2007). If the Amazon droughts of the last decade continue into the future, ∼55% of Amazon Basin forests will be damaged, emitting between 15 and 26 Pg C into the atmosphere (Nepstad et al. 2008). Modeling suggests that the accumulation of greenhouse gases (GHGs), and the associated increase in radiative forcing of the atmosphere, will cause a decrease of more than 20% in rainfall across some parts of the Amazon Basin by the end of the twenty-first century (IPCC 2008).

Afforestation of degraded lands

Land degradation is widespread, encompassing ∼24% of the world’s terrestrial area, inhabited by about 1.5 billion people (Bai et al. 2008). Among the many effects of these degradation processes on croplands and grazing lands, is the reduction in vegetation cover and depletion of SOC, resulting in decreased formation and stability of the soil structure (Lal 2002). A combined effect of these processes includes increased raindrop impact on the exposed surface, decreased hydraulic conductivity of the soil, and enhanced loss of the fertile top layer by erosional processes (Stavi & Lal 2011a). The SOC lost through these erosional processes could be oxidized and emitted as CO₂ to the atmosphere (Stavi & Lal 2011b), or buried and accumulated in depositional sites (Gregorich et al. 1998). Regardless, the depletion of SOC from the surface layer further degrades the soil’s fertility and productive capacity (Stavi & Lal 2011a).
Table 1. Impact of wildfire-produced charcoal on soil and trees in forest lands.

<table>
<thead>
<tr>
<th>Location</th>
<th>Climate</th>
<th>Soil type</th>
<th>Tree species</th>
<th>Impact on soil and litter</th>
<th>Impact on trees</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Northern Sweden</td>
<td>Cold</td>
<td>Various</td>
<td>Silver birch, Scots pine</td>
<td>Charcoal stimulated microbial biomass in some instances, and affected litter decomposition</td>
<td>Greater growth of Silver birch, no impact on net growth of Scots pine; in some instances, increased shoot-to-root ratio of both species</td>
<td>Wardle et al. (1998)</td>
</tr>
<tr>
<td>Montana, USA</td>
<td>Temperate</td>
<td>Typic Dystrocteypts</td>
<td>Ponderosa pine</td>
<td>Charcoal mixed with NH$_4$ (1%/99% w/w) increased rate of nitrification in soil</td>
<td>Charcoal mixed with NH$_4$ decreased solution concentrations of plant phenolics</td>
<td>DeLuca et al. (2006)</td>
</tr>
<tr>
<td>Northern Sweden</td>
<td>Cold</td>
<td>Not specified</td>
<td>Scots pine</td>
<td>Charcoal mixed with humus increased mass loss of humus through either respiration or leaching of soluble compounds</td>
<td>—</td>
<td>Wardle et al. (2008)</td>
</tr>
<tr>
<td>Idaho, USA</td>
<td>Temperate</td>
<td>Not specified</td>
<td>Ponderosa pine, Douglas-fir</td>
<td>Charcoal mixed in mineral soil augmented abundance of ammonia-oxidizing bacteria, and increased nitrification rates</td>
<td>—</td>
<td>Ball et al. (2010)</td>
</tr>
<tr>
<td>Switzerland</td>
<td>Temperate</td>
<td>Cambisol</td>
<td>Various</td>
<td>Charcoal mixed with different organic materials in mineral soil did not increase decomposition rate of litter</td>
<td>—</td>
<td>Abiven and Andreoli (2011)</td>
</tr>
<tr>
<td>Eastern Russia</td>
<td>Cold</td>
<td>Brown taiga</td>
<td>Gmelin larch (Larix gmelini), Scots pine</td>
<td>Charcoal increased the soil’s pH, water content, and available P</td>
<td>Charcoal increased germination and growth of Scots pine, but not that of Gmelin larch</td>
<td>Makoto et al. (2011)</td>
</tr>
<tr>
<td>Victoria, Australia</td>
<td>Temperate</td>
<td>Not specified</td>
<td>Various</td>
<td>Not tested</td>
<td>Not tested</td>
<td>Santín et al. (2012)</td>
</tr>
</tbody>
</table>
Wherever prevailing terrestrial and climatic conditions allow, degraded lands can be converted to afforestation projects. This could be relevant for most of the globe’s biomes. Also, afforestation of degraded lands could improve ecosystem services and enhance biodiversity conservation (Chazdon 2008). At present, afforestation projects take place in many countries and across several climatic regions. Millions of smallholder farmers throughout the world are already engaged in tree planting and forest management to compensate for the loss of access to woody material and the degradation of natural forests (Boyd et al. 2005). In his essay on forestation of degraded lands, Chazdon (2008) stressed that new forests will require adaptive management as resilient systems that can withstand stresses of climate change, habitat fragmentation, and other disturbances.

Several studies have revealed the capability of biochar application to restore degraded lands (e.g., Sohi et al. 2009; Stavi 2012). It is proposed that some of the biochar-amended degraded lands could be used for afforestation. Since biochar augments the retention of water and nutrients in the uppermost soil layer and increases their availability for plant uptake (Lehmann 2007), trees planted in the biochar-amended lands are expected to experience favorable conditions, which would foster their productivity. Also, the alkaline nature of the biochar is expected to alter the soil pH. This may be advantageous in highly weathered soils, where biochar may neutralize their acidity (Lehmann & Rondón 2006). Therefore, it is anticipated that the increased productivity of trees will augment the land’s restoration capacity.

The method of biochar application could be through spreading throughout the land surface, and then either incorporating into the soil, or retaining it on the ground. However, while retention of biochar on the intact ground surface prevents the disturbance of the uppermost soil layer and therefore minimizes the risk of soil erosion, it also decreases the potential improvement of soil nutrient status. Also, the light-weight biochar retained on the surface may be susceptible to redistribution over the surface through wind or water force. Therefore, wherever possible, biochar should be incorporated into the soil. Several types of machinery can be used to spread biochar on the ground surface. Yet, so far, only few spreaders have been developed specifically for biochar (or wood ash) (e.g., Cox 2012; http://www.eng.asu.edu/vital.se/machines.htm). Until such spreaders become more widespread, biochar spread over extensive lands could be conducted through modified conventional spreaders—traditionally used to spread compost, manure, or sludges. Small-scale application of biochar into the soil matrix could take place by using hand-operated rotator machines. Applications over extensive land areas could be carried out with tractor-drawn ploughs, disk ploughs, or rotavators. In these cases, the application depth would be determined in accordance with the terrestrial conditions and tree species.

Ideally, the application of biochar should take place before the trees are planted. On these occasions, the biochar could be applied throughout the project’s area. In the cases of already established afforestation projects, biochar could be applied only in the trees’ inter-rows. Furthermore, when calculating the biochar application rate of either new or already established afforestation projects, the terrain’s surface conditions should be taken into account as they considerably impact the accessible land area. For example, steep terrains may impose difficulties in either manual or mechanized application of biochar. Another example is exposed bedrocks, or bedrocks in very shallow depths, that restrict the application of biochar in soil.

Whether conducted for co-production of bio-energy and biochar, or solely for the production of biochar, the pyrolysis feedstocks could be diverse. Similarly to other bio-energy sectors, feedstocks could include a range of agricultural and organic materials, such as forestry residues, domestic wastes, sewage sludge, and excess livestock manure. Also, designated highly productive agricultural crops that neither require high inputs nor compete with food crops could be used as feedstocks (Tilman et al. 2009). At the same time, residues of agricultural crops, such as wheat straw or corn stalk, must not be considered as relevant feedstocks. This is because the on-site retention of such residues is crucial for maintaining the SOC stock, supporting the soil food web, and protecting the soil surface from erosional processes (Lal & Pimentel 2009).

Established afforestation systems require permanent supervision and occasional maintenance actions such as trimming and thinning. Wherever possible, some of the prunings should be left on the surface, aimed at preventing the emergence of erosional processes, and replenishing the SOC stock (Jones et al. 2008). Under certain conditions, the waste materials are completely cleared from the afforested land in order to reduce the risk of fire outbreaks (Leinonen 2004). One way or another, some of the waste material could be used as feedstock for either bio-energy or biochar production. Ogawa et al. (2006) proposed that in addition to its environmental advantages, the practice of biochar management could also have social and economic benefits through the establishment of new businesses, new job opportunities, and raised income of inhabitants in rural regions. Yet, economic calculations and life-cycle assessments should be conducted on a site-specific basis in order to ensure net economic profit and net C sequestration after the deduction of costs and emissions pertaining to waste collection from the forest floor, transportation to the pyrolysis plant, and processing into end products. Providing specific and effective environmental policies, some of the costs could be covered by the inclusion of this management practice under the international C finance mechanism. This option is further elaborated in the section ‘Regulations and funding’.

Reforestation projects
In a comprehensive research project, Niles et al. (2002) studied the potential C mitigation capacity of reforestation projects across 48 sub-tropical and tropical developing
countries during a 10-year reforestation period between 2003 and 2012. They calculated a reforestation rate of \( \sim 1.7 \), 1.1, and 0.7 million ha year\(^{-1}\) in Latin America, Asia, and Africa, respectively, potentially yielding a total of \( \sim 3.5 \) million ha year\(^{-1}\). The corresponding potential vegetative-C accumulation capacity, excluding the SOC, was calculated as \( \sim 178, 96, \) and 42 Tg, yielding a total of \( \sim 316 \) Tg over the 10-year period. The C stocks of such systems could be increased considerably if biochar is applied to their soil. Similar to afforestation systems, biochar can be produced on its own or as a by-product of the pyrolysis process for production of bio-energy. It is possible that in developed countries, where related infrastructures are well established, this track would be more feasible as a complementary sector to the bio-energy industry. At the same time, in developing countries lacking such advanced industries, the production of biochar would be probably introduced more easily if conducted through low investments in technologies that produce only biochar. Either way, addressing forest residues as a relevant feedstock for bio-energy and biochar production, and utilizing the biochar as soil amendment in reforestation projects, could be perceived as a sustainable alternative to the intentional burning of them. Yet, it is recognized that among many rural populations in developing countries, where fuelwood and home-made charcoal encompass the main source of energy for domestic needs (Mead 2005), little motivation, if any, would be directed at using the charcoal (biochar) as soil amendment. Therefore, in order to induce the desired change in motivation, international policies should regulate the practice of biochar application in forest soil as eligible for funding under the C finance mechanism. It is expected that in conjunction with extension activities aimed at educating local populations about its long-term advantages, this management practice could be extensively implemented.

Also, it is acknowledged that even if the pyrolysis process for bio-energy production, coupled with the use of biochar as soil amendment, could be considered C-neutral, or under optimistic conditions as C-negative (Lehmann 2007), the related procedures of feedstock collection, transportation, processing, and spreading would require utilization of fossil fuel, emitting large amounts of GHGs. Therefore, life cycle assessments of the conversion of residues into bio-energy or biochar should be conducted on a site-specific basis, aimed at verifying the environmental sustainability of this management practice. In order to be considered pertinent, the overall environmental footprint of this management practice should be lower than that under the alternatives (e.g., intentional burning or any other treatment of residues).

In managed and planned reforestation lands, it is preferable that some of the residues would be retained on the forest floor in order to maintain the SOC stock and control erosional processes (Jones et al. 2008). In their review of the potential production of bio-energy from forest wastes, Gregg and Smith (2010) stressed that retention of forest residue should encompass at least 20 Mg ha\(^{-1}\). Using this limitation, any amount of residues beyond that lowest edge could be then used as feedstock for producing bio-energy or biochar. For instance, if the residue rate is \( \sim 80 \) Mg ha\(^{-1}\) (see Gregg & Smith 2010) then the harvested residues could reach up to \( \sim 60 \) Mg ha\(^{-1}\). Considering that the conversion rate of vegetative material to biochar in slow pyrolysis – the thermal decomposition of biomass by slow heating at low to medium temperatures (450–650°C) – is \( \sim 35\% \) (Sohi et al. 2009), then this amount could yield \( \sim 20 \) Mg biochar ha\(^{-1}\). Another widespread practice in some of the temperate and cold regions, including extensive woodlands areas in Finland and the northern USA, is the ‘whole tree method’; where the entire above-ground biomass is being used for either logging or the bio-energy industries, and no residues are left on the ground surface (Leinonen 2004). Considering the heavy environmental cost of this practice, including the rapid diminishment of SOC stocks and the increased risk of soil erosion, it seems that rather than utilizing the residues exclusively for direct production of bio-energy, some of them could be converted to biochar. It is expected that on-site application of the produced biochar in the deforested lands would considerably restore the quality of their soil and improve their restoration capacity, facilitating the establishment of reforestation projects. Yet, even if some of the C is returned to the soil through biochar application, the ‘whole tree method’ is highly un-recommended due to the increased risk of accelerated erosion.

In spite of the apparent benefits of biochar application in forest lands, very little attention has been paid to this management practice so far. For example, in Indonesia, Ogawa et al. (2006) noted that excess wood bark of Acacia mangium stands could be used for biochar production, which would be utilized for several purposes, such as soil water purification methods, and soil amendment in crop-lands and forest lands. In another recent synthesis study, Ogawa and Okimori (2010) reviewed earlier studies of biochar application in the Japanese forestry sector. They showed that bark-biochar powder combined with 0.1–1.0% of various chemical fertilizers, increased growth of both shoot and root of Pinus thunbergii. In addition, this treatment increased yields of the associated edible mycorrhizal fungus Rhizopogon rubescens. Moreover, they showed that the application of biochar combined with chemical phosphate (P) and different mycorrhizal fungi augmented the survival rate of seedlings from several pine species. In a greenhouse study, Bell and Worrall (2011) examined the impact of biochar on a humified organic soil obtained from a British temperate forest. The biochar was applied at rates of 6.25, 62.5, and 87.5 Mg ha\(^{-1}\), and compared to control, non-amended soil. They reported that biochar treatments had no effect on net ecosystem respiration or on nitrate (\( \text{NO}_3^- \)) flux and dissolved organic C (DOC) concentration in the soil leachate. However, soil pH was reported to respond positively to the biochar rate.

Despite scant reporting on the utilization of biochar in forest lands, the use of wood ash derived from biomass burning in power plants has been intensively studied (Table 2). Specifically, utilizing wood ash as soil amendment in forestry systems offers a means to counteract soil
Table 2. Reforestation lands amended with wood ash derived from biomass burning in power plants.

<table>
<thead>
<tr>
<th>Location</th>
<th>Climate</th>
<th>Soil type</th>
<th>Stand age and tree species</th>
<th>Ash source and rate</th>
<th>Impact of ash on soil and trees</th>
<th>Reported impediments</th>
<th>Recommendation</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Southern Finland</td>
<td>Cold</td>
<td>Haplic Podzol</td>
<td>31 to 75-year-old stands of Norway spruce and Scots pine</td>
<td>Loose wood ash, 3 Mg ha(^{-1}) (+ N to a rate of 120–150 kg ha(^{-1}))</td>
<td>Increased nutrient concentrations and higher pH in soil; increase in needle nutrient concentrations; slight increase in tree growth after 5 years and no increase after 10 years</td>
<td>No effect of ash with mineral N on soil N concentration; increased Cd concentration in soil; too high pH under certain conditions, negatively impacting understory vegetation</td>
<td>Utilizing pelleted or granulated ash, rather than loose ash</td>
<td>Saarsalmi et al. (2004)</td>
</tr>
<tr>
<td>Northern Finland</td>
<td>Cold</td>
<td>Haplic podzol</td>
<td>60-year-old Scots pine stand</td>
<td>Wood-, bark-, and peat ash to rates of 1, 2.5, and 5 Mg ha(^{-1}) (+ 185 kg N ha(^{-1}))</td>
<td>Increased nutrient concentrations and higher pH in soil; positive impact on tree growth only for the biochar rates of 2.5 or 5 Mg ha(^{-1}) with mineral N</td>
<td>No effect of ash with mineral N on soil N</td>
<td>—</td>
<td>Saarsalmi et al. (2006)</td>
</tr>
<tr>
<td>Northern Spain</td>
<td>Temperate</td>
<td>Humic Umbrisol</td>
<td>5-year-old Douglas-fir stand</td>
<td>Wood-bark ash, 10 and 20 Mg ha(^{-1})</td>
<td>Increased soil pH and available Ca, Mg, and K; increased foliar K, Ca, N, and P concentrations, and greater tree height, diameter, and biomass</td>
<td>Moderate liming and fertilizing capacity of the ash</td>
<td>Ash application at rates adapted to the liming needs of soil</td>
<td>Solla-Gullón et al. (2006)</td>
</tr>
<tr>
<td>Nordic countries</td>
<td>Cold</td>
<td>Various (review)</td>
<td>Various (review)</td>
<td>Various (review)</td>
<td>Increased soil K, Na, SO(_4), Ca, and Mg in the short-term, and increased soil pH in the long term; almost no impact on trees in mineral soils, and a positive impact on trees in organic soils</td>
<td>In mineral soils, N is the most limiting nutrient for tree growth, decreasing biochar impact</td>
<td>Loose rather than aggregated ash; low rather than high rate doses; for adult rather than young stands; in acidic rather than alkline soils</td>
<td>Augusto et al. (2008)</td>
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(Continued)
Table 2. Continued.

<table>
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<th>Location</th>
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<th>Stand age and tree species</th>
<th>Ash source and rate</th>
<th>Impact of ash on soil and trees</th>
<th>Reported impediments</th>
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<td>Northern Spain</td>
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<td>Solla-Gullón et al. (2008)</td>
</tr>
<tr>
<td>Southern Finland</td>
<td>Cold</td>
<td>Haplic podzol</td>
<td>45-year-old Norway spruce stand</td>
<td>Wood burnt-bark ash, 3 Mg ha(^{-1}) (+ 150 kg N ha(^{-1}))</td>
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<td>Southern Finland</td>
<td>Cold</td>
<td>Haplic podzol</td>
<td>31-year-old stand of Scots pine and 45-year-old stand of Norway spruce</td>
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<td>−</td>
<td>−</td>
<td>Saarsalmi et al. (2010)</td>
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<tr>
<td>North-west Spain</td>
<td>Sub-humid</td>
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<td>31-year-old Monterey pine stand</td>
<td>Mixed wood ash, 7.5 Mg ha(^{-1})</td>
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<td>Decreased P availability; decreased N availability</td>
<td>Biochar combined with P fertilizer</td>
<td>Santalla et al. (2011)</td>
</tr>
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</table>
acidification and to compensate for nutrient loss resulting from intensive logging practices, such as whole-tree harvesting (Saarsalmi et al. 2004). For example, in northwestern Spain, Santalla et al. (2011) examined the impact of mixed wood ash, applied at a rate of 7.5 Mg ha$^{-1}$, on the soil nutrient status of an intensive forest system planted with Monterey pine (Pinus radiata D. Don). In this subhumid Mediterranean region, the A horizon of the soil is rich in organic matter, highly acidic, and with low available nutrients, including P, calcium (Ca), magnesium (Mg), and potassium (K). Santalla and colleagues demonstrated that the application of ash restored soil reserves of Ca, Mg, and K in the soil’s organic layer. This is in accord with Saarsalmi et al. (2004), who noted that the forest humus layer acts as a trap for many of the elements added in wood ash, resulting in a long-lasting, nutritional effect of the ash in the soil. Yet, Santalla et al. (2011) also stated that the application of wood ash decreased the availability of P. They attributed this effect to the charcoal content of ash, which temporarily reduced the solubility of mineral P. They added that this effect could have a specifically adverse impact on acidic soils in temperate forests, where P is the most limiting factor. They also reported that the application of wood ash combined with mineral P fertilizer at rates of 7.5 Mg ash ha$^{-1}$ and 65 kg P ha$^{-1}$, increased P concentration in needles, litterfall, and soil. In addition, Santalla and colleagues indicated that wood ash decreased soil N mineralization rate and mineral N concentrations. They attributed this effect to the greater microbial activity in response to the better nutrient availability, along with the effect of the very high C : N ratio in the ash. They noted that despite not limiting primary productivity under the studied conditions, this impact may adversely affect tree growth where decreased N availability could induce N deficiency in plants. This effect is supported by other studies (Augusto et al. 2008; Saarsalmi et al. 2010) which demonstrated that in mineral soils and in infertile sites, growth is mainly dependent on N availability. Saarsalmi et al. (2004) commented that since on poor mineral soils the growth response to wood ash alone seems to be insignificant or even declined, the direct economic benefit of this management practice is negligible. Conversely, Solla-Gullón et al. (2008) showed that ash application in N-rich mineral soils of temperate areas significantly improved tree growth.

The impact of wood ash combined with mineral N on plant nutrient availability and tree growth has been intensively studied. For example, in southern Finland, 3 Mg ha$^{-1}$ of wood burnt-bark ash combined with 150 kg N ha$^{-1}$ applied in the soil of a 45-year-old Norway spruce (Picea abies (L.) Karst.) was reported to increase pH, base saturation, and total and extractable concentrations of nutrients including P, K, Mg, and Ca in the soil’s organic layer (Helmisaari et al. 2009). Similar effect was reported by Saarsalmi et al. (2010) who studied, in the same geographic region, the impact of burnt-bark ash combined with mineral N in a 31-year-old Scots pine stand. However, Helmisaari et al. (2009) reported that during a 10-year period, ash mixed with mineral N reduced the biomass of fine roots to $\sim$30% less than that under soil not amended with either ash or mineral N. At the same time, this treatment increased K concentrations in needles, but overall tree growth was similar between this treatment and the non-amended soil. In a study in 31- and 75-year-old coniferous stands in southern Finland, Saarsalmi et al. (2004) reported that wood ash alone had no effect on the P, K, and Ca concentrations in the needles, but ash combined with N fertilizer increased concentrations of these nutrients in the needle. Overall, they reported no impact on tree growth under ash alone, when compared with a slight increase in tree growth under ash combined with N. That slight impact appeared during the first 5 years after application and then diminished. However, Saarsalmi et al. (2006) reported a positive impact of biochar combined with mineral N on growth of Scots pines even 23 years after the application.

Above all, the key determinants of wood ash chemistry are the tree species combusted, the nature of the burning process, and the prevailing conditions at the application site. For example, wood ash from hardwood species contains higher levels of macronutrients in their ash than conifers, and their silica (SiO$_2$) content is usually lower. With regard to the conditions of burning, a furnace temperature of between 500$^\circ$C and 900$^\circ$C is critical for the retention of nutrients, particularly K. This temperature also determines the concentrations of potentially toxic components in the ash (Pitman 2006). Also, the mode of ash application seems to have a considerable impact on nutrient availability for trees. For example, solubilization of wood-bark ash could be low when spread on the ground surface without being incorporated into the soil (Solla-Gullón et al. 2006). Conversely, incorporation into the soil was reported to increase solubility of wood ash compounds, augmenting soil fertility and increasing tree growth (Solla-Gullón et al. 2008). In addition, the size of ash particles is also expected to play a role in impacting soil features. For instance, loose ash material was reported to have greater reactivity than pelleted or granulated ash (Saarsalmi et al. 2004), accelerating the rate of release of Ca, K, and Na (Pitman 2006). Therefore, ashes of certain particle size and specific surface area could assist in mitigating undesired high acidity of soils.

Additionally, one of the major concerns regarding wood ash is that it might increase concentrations of heavy metals in soil. For example, the application of wood ash was reported to increase cadmium (Cd) concentration in the soil’s organic layer in several forests (e.g., Saarsalmi et al. 2004). In this regard, Pitman (2006) commented that fly ash – the lightest component that accumulates in the flue system – can contain high concentrations of heavy metals such as Cd, copper (Cu), chromium (Cr), lead (Pb), and arsenic (As) and that this ash should not be used as soil amendment. However, Solla-Gullón et al. (2008) noted that mixed wood ash, comprised of fine fly ash, and bottom wood ash is much less reactive and contains lower amounts of trace elements than fly wood ash alone, reducing the risk of heavy metal contamination. Finally, it is yet questionable...
whether higher application rates of wood ash augment the potential productivity of forestry systems (Saarsalmi et al. 2006), or increase the risk of toxic effects (Pitman 2006). In this regard, it is important to stress that, depending on feedstock origin and pyrolysis conditions, biochars may also contain heavy metals, as well as other contaminants. However, the ecotoxic effects of these contaminants can be avoided by testing the feedstocks before pyrolysis and examining the biochars before they are applied to soil (Ernsting 2011). Yet, there is great need to further assess these mechanisms, as well as identify specific feedstock characteristics and operational conditions which lead to the formation and retention of these contaminants in the final biochar product (Verheijen et al. 2010).

The cost of converting forest residues into bio-energy or biochar varies greatly and depends on a wide range of variables. These include the terrain’s features, climate conditions, fuel price, utilized technology for harvesting, collection and processing of feedstocks, the use of transportation means, and distance to the power plants. Since a major economic constraint lies in the costs associated with transportation (Leinonen 2004), the conversion of residues into bio-energy and biochar is best conducted on-site. Therefore, the construction of pyrolysis plants for bio-energy production, or merely kilns for production of biochar, should be conducted as close as possible to the reforested lands (Matovic 2011). Either way, economic calculations for the production of bio-energy or biochar should be implemented on a site-specific basis and encompass all costs related to the processing of feedstocks. Similar to afforestation projects, also in reforestation lands the application of biochar should ideally be conducted before planting the trees, augmenting the biochar application capacity of the land unit, and increasing the system’s potential production capacity. Also, similar to afforested lands, physical restrictions in the terrain’s surface should be considered when calculating the biochar application capacity.

Overall, it seems that the impact of wood ash on soil characteristics and stand productivity are site-dependent, vary greatly on the temporal axis, and are affected by the ash features. On the whole, to date, most of the studies which have examined the impact of wood ash in forestry systems were implemented under humid and temperate-to-cold regions, on acidic soils, and for coniferous stands. At the same time, research under Mediterranean or drier and hotter regions, on alkaline soils, and for broadleaf stands is extremely scarce.

**Multi-purpose agroforestry**

Agroforestry systems – agricultural lands with more than 10% tree cover – occur on 46% of global agricultural land area, and affect 30% of the world’s rural population. Therefore, this land use represents over 1 billion ha of land and 558 million people worldwide and is particularly prevalent in Southeast Asia, Central America, and South America, with over 80% of agricultural land area under agroforestry (Zomer et al. 2009). This land use is a multi-purpose practice and, according to the landowner objectives, can involve any combination of timber, row crop, pasture and forage, fruit crop, fuelwood, wildlife habitat, or recreational sites (Workman & Allen 2004). Specific agroforestry practices include alley cropping, multi-storey cropping, or silvopastoral systems, where trees or shrubs are intercropped with grain crops, vegetables, or forages (Kandji et al. 2006). The trees themselves can be multi-purpose, for example, providing fruits, producing small-diameter wood for construction, supplying fuelwood or stakes (Boyd et al. 2005), or be used for hedging, shade, ornamentals, or medicinal plants (Maroyi 2009). The multiple products of an agroforestry system are available at different time intervals, can utilize space more effectively, and extract nutrients from different soil layers more efficiently. The diversity of products help farmers to cope with crop failure (Workman & Allen 2004) and can raise household incomes (Maroyi 2009). Such systems in the humid tropics are part of a continuum of landscapes ranging from primary and managed forests to croplands and grasslands (Kandji et al. 2006). Overall, agroforestry systems were reported as being capable of sequestering between 15 and 228 Mg C ha$^{-1}$ (Kandji et al. 2006). Average C storage by agroforestry systems has been estimated by Montagnini and Nair (2004) as 9, 21, 50, and 63 Mg C ha$^{-1}$ in semi-arid, sub-humid, humid, and temperate regions, respectively. For smallholder agroforestry systems in the tropics, potential C sequestration rates range between 1.5 and 3.5 Mg C ha$^{-1}$ yr$^{-1}$. Nevertheless, they stressed that while agroforestry systems with perennial crops may be efficient C sinks, intensively managed agroforestry systems with annual crops are more similar to conventional agricultural lands in terms of net C fluxes. Moreover, they added that even the perennial crop-tree combinations do not attain C sequestration capacities comparable to natural forests.

Agroforestry systems not only provide a great opportunity for sequestering C, and hence help mitigate climate change, but they also enhance the resilience of agricultural systems and increase additional ecosystem services. Such systems can efficiently enlarge water use efficiency, provide shading, increase nutrient turnover, and improve micro-habitat conditions for agricultural crops, increasing their adaptation capacity to climatic changes. Moreover, Kandji et al. (2006) showed that in the event of leguminous trees, agroforestry systems can provide N to crops, reducing input on fertilizers. However, such trees may increase emissions of nitrous oxide (N$_2$O) compared with unfertilized systems. Yet, it is expected that N-fertilizer amended soils emit N$_2$O to at least the same extent as legume tree-based agroforestry systems. In addition, compared with mono-species croplands, agroforestry systems may have better resilience to pest infestation. As shown by Paul et al. (2010), increased plant biodiversity due to mixing trees and herbaceous species in agricultural landscapes, results in positive interactions that assist in controlling outbreaks of pests and diseases. As stressed by Stavi & Lal (2013),
an additional ecosystem service provided by agroforestry systems is the better erosion control, sustaining the on-site quality of soil and off-site quality of water sources.

Relevant feedstocks for biochar production in agroforestry systems could be the tree prunings. Nevertheless, it is expected that emissions of GHGs associated with the collection of prunings, transportation to the pyrolysis plant, and processing would decrease net amounts of C sequestration. Yet, it should be acknowledged that some practices (e.g., taking the prunings out of the field) will be carried out in either case and, therefore, have to be deducted from the biochar-track’s life cycle assessments. Moreover, since many farmers routinely burn prunings in order to dispose them, the conversion to biochar could be considered highly sustainable. In the cases of mixed farms that include land cultivation and livestock raising, the excess livestock manure could also be used for this purpose. However, residues from the main (inter-tree) crops must not be considered as viable feedstocks because of their important role in preserving the quality and productive capacity of the soil in the inter-row spaces.

It is expected that the application of biochar in the soil of agroforestry systems would decrease leaching of nutrients, reducing contamination of below-ground water sources. Also, the increased stability of the surface soil would reduce the generation of water overland flow and decrease frequency and magnitude of erosional processes. These effects are expected to lessen eutrophication of above-ground water sources and reduce siltation in reservoirs. The application of biochar was also reported to decrease emissions of N$_2$O and methane (CH$_4$) from soil (Rondon et al. 2005). The expected increased fertilizer efficiency in the biochar-amended soil (Lehmann 2007) would enable lowered fertilizer rates, decreased production costs, and reduced off-site and on-site emissions of GHGs. A recent study by Stavi & Lal (2013) highlighted the high capacity of the practices of agroforestry systems and biochar application in croplands – as opposed to other agricultural practices – in offsetting climate change. It is hereby suggested that the combination of these two practices in the same land unit would link their climate change mitigation and adaptation capacities, further increasing their agricultural and environmental benefits.

Mono-culture of fruit tree orchards and bio-energy plantations

Similar to afforestation, reforestation, and agroforestry systems, the application of biochar in soil could also be a relevant management practice in mono-culture fruit tree orchards and bio-energy tree plantations. Also, similar to other tree-based agro-ecosystems, the biochar would be preferably applied to the whole area before planting the trees, maximizing C sequestration capacity, and augmenting the system’s resilience. In the event of already established orchards or plantations, biochar could be applied only in the inter-row spaces.

All the relevant feedstocks for other tree-based agro-ecosystems could also be pertinent for fruit tree orchards and bio-energy plantations. In addition, prunings from the orchards and plantations could also supply considerable amounts of feedstocks. For example, in the Veneto region of northern Italy, vines (Vitis spp.), olive trees (Olea europaea L.), apple trees (Malus domestica borkh.), pear trees (Pyrus spp.), peach trees (Prunus persica (L.) Batsch), citrus trees (Citrus spp.), almond trees (Prunus dulcis (Mill.) D.A. Webb), and hazelnut trees (Corylus spp.) were reported, respectively, to produce 2.53–2.90, 1.70–2.00, 0.15–2.40, 0.09–2.00, 0.20–2.90, 0.32–1.80, 0.15–1.70, and 0.19–2.80 Mg prunings ha$^{-1}$ year$^{-1}$ (Santacroce 2010). In sub-tropical Asia, coconut trees and cocoa trees were reported, respectively, to produce 5–10 and $\sim$25 Mg residues ha$^{-1}$ year$^{-1}$ (Koopmans & Koppejan 1997). In the Middle East, date palm orchards produce $\sim$7.5 Mg dry fronds ha$^{-1}$ year$^{-1}$ (personal communications, Stavi). Similar to agroforestry systems, also in fruit tree orchards and bio-energy plantations, emissions of GHGs are expected during collection, transportation, and processing of the prunings. Yet, as with agroforestry systems, the collection of prunings is inevitable in the maintenance of many of the orchards and plantations. Specifically, for the date palm orchards, where many farmers burn the fronds in order to dispose them (personal communications, Stavi), their conversion to biochar impose net environmental profit. Tree plantations for bio-energy cropping produce smaller, similar, or greater amounts of feedstock. For instance, across the USA, poplar (Populus spp.) and willow (Salix spp.) are extensively cropped for bio-energy, producing, respectively, between 1.44–1.88 and 1.82–1.98 Mg of dry biomass ha$^{-1}$ year$^{-1}$ (De La Torre Ugarte et al. 2003). In the sub-Mediterranean region of Spain, 3-year harvest rotations of eucalyptus (Eucalyptus spp.) were reported to produce much greater yields of between 45 and 60 Mg dry biomass ha$^{-1}$ at the third and sixth years after planting (Tolosana et al. 2010).

Similar to other sources of feedstocks, assuming a conversion rate from biomass to biochar of $\sim$1:3 (Sohi et al. 2009) and considering the application of the minimal recommended rate of 10 Mg ha$^{-1}$ (Chan et al. 2007), highly productive bio-energy crops, such as eucalyptus, could easily provide a sufficient amount of biochar for on-site application during a single harvest. Using the same calculations, the on-site utilization of biochar produced from other bio-energy crops and fruit tree prunings could be conducted gradually along the temporal axis. Otherwise, biochar application in the soil of these crops could rely on off-site sources.

Early evidence of biochar (then called charcoal) use as a soil amendment in fruit trees was indicated for Satsuma mandarin (Citrus unshin Marc.) trees on trifoliate orange (Poncirus trifoliate Raf.) rootstocks in Japan. Biochars produced from rice husk, citrus juice sediments, or spruce bark were reported to increase root length, enlarge shoot productivity, augment NPP, and stimulate development of vesicular arbuscular mycorrhiza (Ishii & Kadoya 1994).
In addition to the impact on tree productivity, and similar to agroforestry systems, the application of biochar in orchards and plantations is expected to decrease their environmental footprint by reducing fertilizer rates. It is anticipated that the outcome will be a reduction in direct and indirect emissions of GHGs and lowered the levels of pollution in above- and below-ground water sources.

Regulations and funding

The world’s focus on mitigating C emissions is progressively increasing. Specifically, international projects in developing countries are concentrating on the slowing of tropical deforestation and promotion of reforestation projects (Niles et al. 2002). An efficient economic mechanism— the C finance mechanism—allows countries to trade in emissions, where developing countries sell C credits aimed at assisting developed countries to achieve emission targets. Such mechanisms could potentially generate considerable funding for forestry projects in the developing world (Miles & Kapos 2008). Yet, key requirements needed to curtail deforestation include strengthened technical and institutional capacities in many developing countries, agreement on a rigorous system for measuring and monitoring emission reductions, and de facto commitment by developed countries to create demand for C credits (Gullison et al. 2007). Also, in many developing countries, one of the major challenges is to maintain a balance between meeting increased market demand for forest products, alleviating poverty, and protecting natural resources. These three conflicting needs are meant to be addressed by small-scale, offset projects under Article 12 of the Clean Development Mechanism (CDM) of the Kyoto Protocol (1998). The objective of this framework is to improve livelihood conditions in developing countries by providing a market mechanism aimed at mitigating climate change through sustainable use of natural resources (Boyd et al. 2005), such as forestry systems.

Yet, the CDM has very strict rules for participation that may be beyond the reach of small-scale farmers. There is a need for institutional support at the national, regional, and international levels to facilitate effective participation of these farmers in forestry projects (Kandji et al. 2006). Also, to ensure the successful implementation of such projects, the local social, cultural, and political context must be comprehensively addressed. In addition, active participation in decision-making by local stakeholders and community-based organizations, free access to information by the local populations, and local ownership of different project components are crucial (Boyd et al. 2005). Moreover, CDM payments should be high enough to encourage individuals to implement forestry projects. For example, Conant (2011) showed that a price of $US 75 Mg $^{-1}$ C would generate such motivation, whereas a lower price would serve as a disincentive for the initiation of such projects. Sohngen and Sedjo (2006) calculated that prices ranging between $US 100–800 Mg $^{-1}$ C could lead to global sequestration of 48–147 Pg C in forestry lands by the end of the twenty-first century.

Following the 13th Conference of the Parties of the United Nations Framework Convention on Climate Change (UNFCCC [2007, Bali]), the parties were encouraged to seek further innovative and efficient means to mitigate climate change. One of the main mechanisms suggested was the Reducing Emissions from Deforestation and Degradation (REDD) of forest lands in developing countries, aimed at maintaining forest C stocks by slowing or eliminating anthropogenic-induced disturbances (Seymour and Forwand 2010). The UN-REDD program supports countries to develop capacities for reducing emissions from deforestation and forest degradation and to implement future mechanisms in a post-2012 climate regime (Verchot and Petkova 2010). This development has resulted in the establishment of many public and private REDD initiatives at local, national, and global levels. Nonetheless, to be effective, REDD efforts will not only have to reverse the economic incentives that drive forest loss but will also need to clarify land tenure, lead international endeavors in limiting illegal logging and trade, and manage trade-offs among competing needs (Seymour & Forwand 2010). The main factors determining the success of REDD projects include: clearly formulated design; governance and land tenure; equity and transparency; indigenous peoples’ rights and knowledge; local–international coordination; and local and institutional capacities (Dulal et al. 2012). Successful implementation of REDD mechanisms would generate a number of gains, including: (1) environmental benefits, such as biodiversity protection, soil and water conservation, and ecosystem restoration; (2) social benefits related with sustainable development and poverty reduction; and (3) governance benefits associated with improved protection of human rights (Verchot & Petkova 2010).

It is proposed that alongside decreases in deforestation rates and increases in reforestation and afforestation efforts, biochar management practice in forestry and tree-based agro-ecosystems ought to be considered as a legitimate strategy for C sequestration. In this regard, it must be emphasized that the production of biochar—either on its own through low-tech stoves or to a much larger extent as a by-product of the pyrolysis process for production of bio-energy—requires investments in apparatus and infrastructure. It is expected that, unless biochar is considered valuable merchandise, most landowners and users, being either individuals, commercial companies, NGOs, or others, will not invest in its production and utilization. As such, financing to facilitate the development of biochar projects in forestry and tree-based agro-ecosystems must be regulated. It is therefore important to emphasize that the application of biochar in soil is not clearly eligible for funding under the CDM (Whitman and Lehmann 2009). To address this, an inter-governmental intervention should adjust the C finance mechanism to explicitly include the biochar management practice aimed at augmenting C sequestration in soil, while increasing vegetative
productivity and reducing environmental footprint of land-use change. A progression such as this would enable a valuation of biochar application in forestry and tree-based agro-ecosystems according to the price given for offsetting of CO₂ emissions.

In a comprehensive life cycle assessment of pyrolysis of different feedstocks and the application of the biochars in croplands, Roberts et al. (2010) concluded that these systems’ viability is dependent on the costs of feedstock production and pyrolysis process, as well as on the value of C offsets. Roberts and colleagues calculated that in order to be profitable, a minimum CO₂ offset price would need to be $US 40 Mg⁻¹ for corn stover, $US 62 Mg⁻¹ for switchgrass, and only $US 2 Mg⁻¹ for yard waste. They demonstrated that livestock manure may also be promising for biochar production. As discussed above, forestry waste and residues derived from fruit tree orchards and bio-energy plantations may also be viable as feedstocks for biochar production.

It must be stressed that, unlike the CDM or REDD programs that have been applied exclusively for developing countries, the use of biochar in soils of forestry and tree-based agro-ecosystems should be regulated for the developed world as well. Further, it is suggested that as long as regulations for biochar projects remain unresolved, payments should be given by authorities to landowners to accomplish improvements in related ecosystem services. The most relevant ecosystem services are C sequestration, soil erosion control, preservation of off-site water source quality, and increased biodiversity. An additional environmental benefit stems from the fact that the application of biochar in tree-based systems is expected to augment their wood and food product yields, lessening motivation for further deforestation. Extensive research is required to investigate the potential of biochar application in soil of forestry and tree-based agro-ecosystems, with a goal of improving their climate change mitigation and adaptation capacities. This should include field experiments under a range of geographical settings, as well as extrapolations and modeling studies on regional and global scales.

**Integration and conclusions**

Natural sequestration of C in forestry and tree-based agro-ecosystems occurs in a relatively efficient manner. Yet, the amounts of C sequestered in these systems are smaller than in primary forests. Moreover, trees’ biomass-C and organic-C in the soil of these systems are not safely sequestered, and large parts of them sooner or later decompose, emitting enormous quantities of CO₂ to the atmosphere. Applying biochar to the soil of such systems would considerably increase the non-decomposable fraction of C in their soil, augmenting their capacity to offset concentrations of atmospheric CO₂. If considering an application rate of between 10 and 100 Mg biochar ha⁻¹ (similar to that in croplands: Chan et al. 2007), and knowing that biochar-C concentration ranges between 50% (Lehmann et al. 2006) and 78% (Gaskin et al. 2008), and assuming that the decomposable portion of biochar-C is up to 20% (Ogawa et al. 2006), this management practice could sequester between 4 and 62 Mg biochar-C ha⁻¹ over the long run. In a recent synthesis study, lands suitable for afforestation and reforestation were reported to encompass 24, 333, 195, 93, 63, and 41 million ha in Central America, Southern America, sub-Saharan Africa, East Asia, South Asia, and Southeast Asia, respectively. Therefore, assuming a global cover of ~750 million ha of these lands (Zomer et al. 2008), and an additional 1 billion ha of agroforestry systems (Zomer et al. 2009), the total relevant land area could comprise of 1.75 billion ha. Applying biochar to these lands could potentially sequester up to a maximum of 109.2 Pg biochar-C (Figure 1). However, as emphasized above, the available area may be smaller due to certain terrestrial conditions such as steep slopes or rocky surfaces. Additionally, wherever forestry lands are subject to extreme, harsh terrestrial conditions, it may be impossible to apply biochar mechanically. Under such conditions, manual application would only be possible to the planting spots, where it would merely be mixed with the soil to be filled in the planting pits. Nevertheless, this limitation would presumably not occur in agroforestry systems, where the trees’ inter-row spaces – assumed to cover 50% of the ground surface – are

![Figure 1. Potential biochar-C sequestration capacity in afforestation, reforestation, and agroforestry systems around the world. Note: For clarity purposes, only 8 tracks are presented. The total number of possible tracks is 16, all of which lie in the range between 2 and 109.2 Pg.](image_url)
typically cultivable. Therefore, if consider only the already established agroforestry systems, the minimum biochar-C sequestration capacity would be of 2 Pg. An additional considerable amount could be sequestered in the soil of the world’s fruit tree orchards and bio-energy plantations.

The biochar management practice could improve soil quality, increasing the resilience of forestry and tree-based agro-ecosystems and promoting their adaptation capacity to the changing climatic conditions. Yet, it could be expected that the impact of biochar on soil and trees would be site-dependent. Therefore, different types of biochars and different application rates may be preferred for diverse geographic regions. Also, combining biochar with mineral N or P or with livestock manure may increase nutrient availability for trees, augmenting their productive capacity. Yet, so far, most of the relevant studies have not dealt directly with biochar, but rather with charcoal produced in wildfires, or wood ash derived from biomass burning in power plants. Extensive research is therefore needed to increase our understanding of the effects of biochar in forestry and tree-based agro-ecosystems. Specifically, the potential ecotoxic effects of biochars should be addressed in order to reduce adverse impacts on human health and the environment. Regulations at international levels are essential in acknowledging this management practice as an eligible strategy for funding under the C finance mechanism, with a goal of promoting its expansion throughout the world.

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